interpreted to be crystalline bedrock, perhaps the Wissahickon Schist mapped locally (Lyttle and Epstein, 1987; Schenck et al., 2000). Nearby grab samples confirmed the presence of a rocky bottom characterized by angular fragments and rounded cobbles, though the actual bedrock geology is unknown. Bedrock exposures at or within 50 cm of the bottom are particularly numerous between Tinicum Island and Chester. The mapped distribution of these exposures is shown in Figure 39.

The paucity of sedimentary cover in Zone 4 reveals that sediment accumulation is negligible on the long term, though there are clear exceptions. Where the bottom has been deepened through dredging, fine-grained sediments trapped within the channel can accumulate to form localized depocenters. Sediment accumulations within the shipping channel are not trivial; indeed, independent estimates (Biggs and Beasley, 1988; USACE, 1973) suggest that the mass of sediment dredged annually from the channel (~$3x10^6$ tons) exceeds that supplied to the estuary by rivers on an annual basis (1–$2x10^6$ tons). Clearly, not only does channel maintenance create a bathymetric trap for sediments, it permanently removes material that would otherwise disperse throughout the open estuary and hydraulically contiguous environments. In this manner dredging constitutes a net sink for sediment in the river–estuary system.

5.6. Radioisotope Profiles and Sedimentation Rates

5.6.1. Reconnaissance Cs-137 Measurements

A total of 25 HDC samples from the subtidal estuary were collected early in this study to evaluate the potential of Cs-137 as a sediment chronometer. Of these, 13 cores from muddy depositional sites were selected for reconnaissance Cs-137 measurements (Figure 40; Table 4). Core subsamples were first counted at low resolution (top, middle, and bottom) as gross measure of sediment "age", because the mere presence/absence of Cs-137 in sediments is an indication of deposition after or before 1954, respectively. Only at three sites, Tinicum Island shoal (C-14b), Marcus Hook East (MHE), and Smyrna River mouth (C-16A), were Cs-137 activities high enough to warrant more detailed measurements. Elsewhere the activities were at or below detection limits (Appendix C), suggesting that net accumulation of mud was locally negligible and (or) the bottom erosional, since 1954. In sum, the reconnaissance measurements revealed that the rates of fine-sediment accumulation between Burlington and New Castle are by and large too

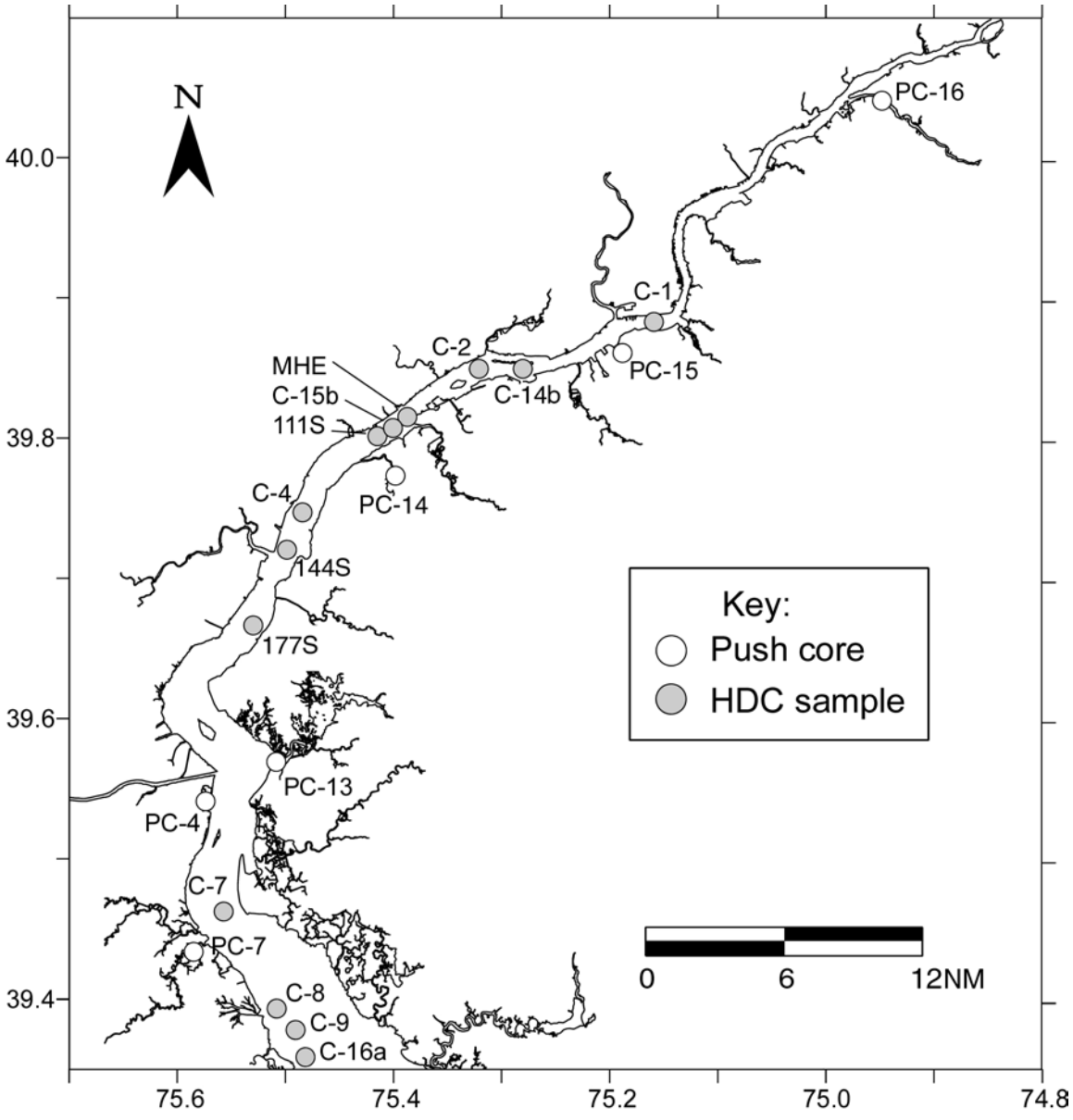

Figure 40. Locations of cores analyzed for radioisotopes.

low, or the bottom too disturbed, for the Cs-137 method to provide reliable chronologies. This result is consistent with sonar observations of a physically reworked bottom down-estuary of Philadelphia. As detailed later tidal marsh and floodplain sites of the upper estuary proved more amenable to Cs-137 dating.

Table 4. HDC sample and push-core locations

Location	Site	Collection Date	Latitude	Longitude
Rancocas Creek	PC-16	7/19/02	40° 02.053	74° 56.116
Woodbury Creek	PC-15	7/18/02	39° 51.040	75° 10.605
Tinicum Island shoal	C-14b	7/20/01	39° 51.023	75° 16.860
Marcus Hook ship channel	C-15b	7/20/01	39° 48.779	75° 23.975
Marcus Hook shoal	MHE	3/22/02	39° 48.627	75° 23.852
Oldmans Creek	PC-14	7/18/02	39° 46.546	75° 24.021
Salem River Marsh	PC-13	4/6/02	39° 34.645	75° 30.240
St. Georges Marsh	PC-4	2/23/02	39° 32.336	75° 34.157
Blackbird Creek	PC-7	4/5/02	39° 26.157	75° 34.924
Smyrna River mouth	C-16A	7/20/01	39° 21.165	75° 28.157

The idealized sediment-depth profile for an impulse tracer like Cs-137 generally mirrors the atmospheric source function and exhibits the following characteristics up-core: (1) a steep limb of increasing activity consistent with the 1954 onset; (2) a prominent peak representative of maximum atmospheric fallout around 1964; and (3) a limb of gradually decreasing activity to non-detectable values in post-1980 sediment intervals (Olsen et al., 1981). For the sedimentary record to faithfully archive the atmospheric source function, the following conditions must be met: (1) steady-steady sediment accumulation; (2) negligible post-deposition mixing of the solid phase (i.e., by burrowing organisms); and (3) chemical immobility of Cs-137 in the sediment column (no desorption). Because steady-state accumulation is atypical for dynamic sedimentary environments, a "1964" peak is not universally present in Cs-137 profiles, and instead the penetration depth of Cs-137 can be used to compute a sedimentation rate averaged since 1954 (Nittrouer et al., 1984). For this approach to be valid, sedimentation post-1954 must be more-or-less continuous, and deep biological mixing negligible. One effect of deep mixing is to increase Cs-137 penetration in the sediment column, thereby increasing the computed sedimentation rate over the true value. Numerical modeling has shown that

the influences of biological mixing (and chemical diffusion) are negligible when the true rate of sedimentation approaches 1 cm/yr (Olsen et al., 1981). Nonetheless, sedimentation rates based on radiotracer profiles should be conservatively viewed as *maximum* estimates when biological mixing is suspected.

None of the cores from the subtidal estuary displayed the ideal Cs-137 profile described above, and complementary Pb-210 profiles exhibited non-steady-state behavior, precluding use of these radiotracers to quantify sedimentation rates (Figure 41). Cesium-137 activity in cores C-14b, MHE, and C-16a was distributed near-uniformly with depth, an indication that deposition is rapid (and likely episodic) at these sites. This type of Cs-137 profile is typical for bottoms subject to intense erosion–deposition cycles or dredging disturbances (Schaffner et al., 1987; Olsen et al., 1993). Consequently, because these Cs-137 profiles might not be complete, the sedimentation rate can be only be constrained to >1 cm/yr.

5.6.2. Seasonal Deposition in the Estuary

Presence of Be-7 activity in two cores collected in spring 2001 shed light on the nature of seasonal deposition in the estuary. In general, occurrence of this radioisotope in bed sediments indicates that its depositional flux exceeds the rate of loss through radioactive decay or physical redistribution. In other words, Be-7 labels sediments that were suspended in the water column (in contact with the Be-7 source) on a timescale within about three half lives or five months of detection. Beryllium-7 was detected to depths of 32 cm and 4 cm in cores C-15b and C-9, respectively (Figure 42), consistent with short-term deposition rates on the order of centimeters per month. On the longer term net accumulation at Site C-15b is non-existent due to dredging, and ≥1 cm/yr at C-9 based Cs-137 occurrence (Figure 42). This observation reveals that although seasonal deposition rates are rapid, subsequent resuspension (or dredging) renders net accumulation rates considerably lower on the long term, a condition observed in other river-estuaries (Hirschberg et al., 1996; Woodruff et al., 2001). Although the actual mechanisms of short-term deposition in the estuary have yet to be elucidated, the Be-7 distributions reveal areas where fine-grained sediments are sequestered on a seasonal basis.

5.6.3. Marsh and Floodplain Sediment Accumulation

Five of the six marsh cores displayed Cs-137 activities that increased upcore and decrease subsequently toward the top (Figure 41). The lone exception was PC-7 (Blackbird Creek), in which Cs-137 activity was highest at the top. Only at Woodbury Creek (PC-15) did Cs-137 display the ideal profile with a prominent "1964" peak. At St. Georges Marsh, presence of Cs-137 to the core bottom suggests that the full inventory was not recovered. Likewise, some fraction of surficial Cs-137 inventory at the Salem Marsh site (PC-13) was lost when dense reed roots were removed to enable coring.

Sediment accumulation rates computed from the Cs-137 profiles following Equation 3 are presented in Table 5. Overall, accumulation rates ranged from 0.3 to 1.5 cm/yr, typical for subtidal–intertidal environments of New Jersey and Delaware. Because biological mixing intensity cannot be quantified from the Cs-137 profiles alone, potential mixing effects on the computed accumulation rates can only be surmised. Following the numerical modeling results of Olsen et al. (1981), at true sedimentation rates of 0.3–5.0 cm/yr and mixing intensities of 0.1–1.0 cm^2/yr, the penetration of Cs-137 is increased by mixing to ≤6 cm below the "1954" datum. Mixing on this scale would cause the computed rates (Table 5) to be overestimated by 10–25 %, but the actual margin of error is apt to be much lower in the present case. Well-preserved bedding surfaces were observed in cores, suggesting that macrofaunal bioturbation is minimal, if present at all. Note that marsh sedimentation rates in general vary with various factors including sediment supply, elevation with respect to tidal inundation, and vegetation type and density. Because only one site at each marsh was sampled in this study, the spatial variability of intra-marsh sedimentation rate is unknown. *Accordingly, the reported sedimentation rate should not be assumed representative for the marsh as a whole.*

A noteworthy observation is that Cs-137 activity is present at significant levels in surficial deposits (core tops), despite the fact that atmospheric fallout has been insignificant since about 1980. Considering that significant post-depositional biological mixing is unlikely, there must be an additional source for Cs-137 in the estuary, perhaps wash-in from the watershed and (or) erosional redistribution of previously buried sediments. As discussed later, radioisotope inventories provide additional insight to the nature of Cs-137 in the estuary.

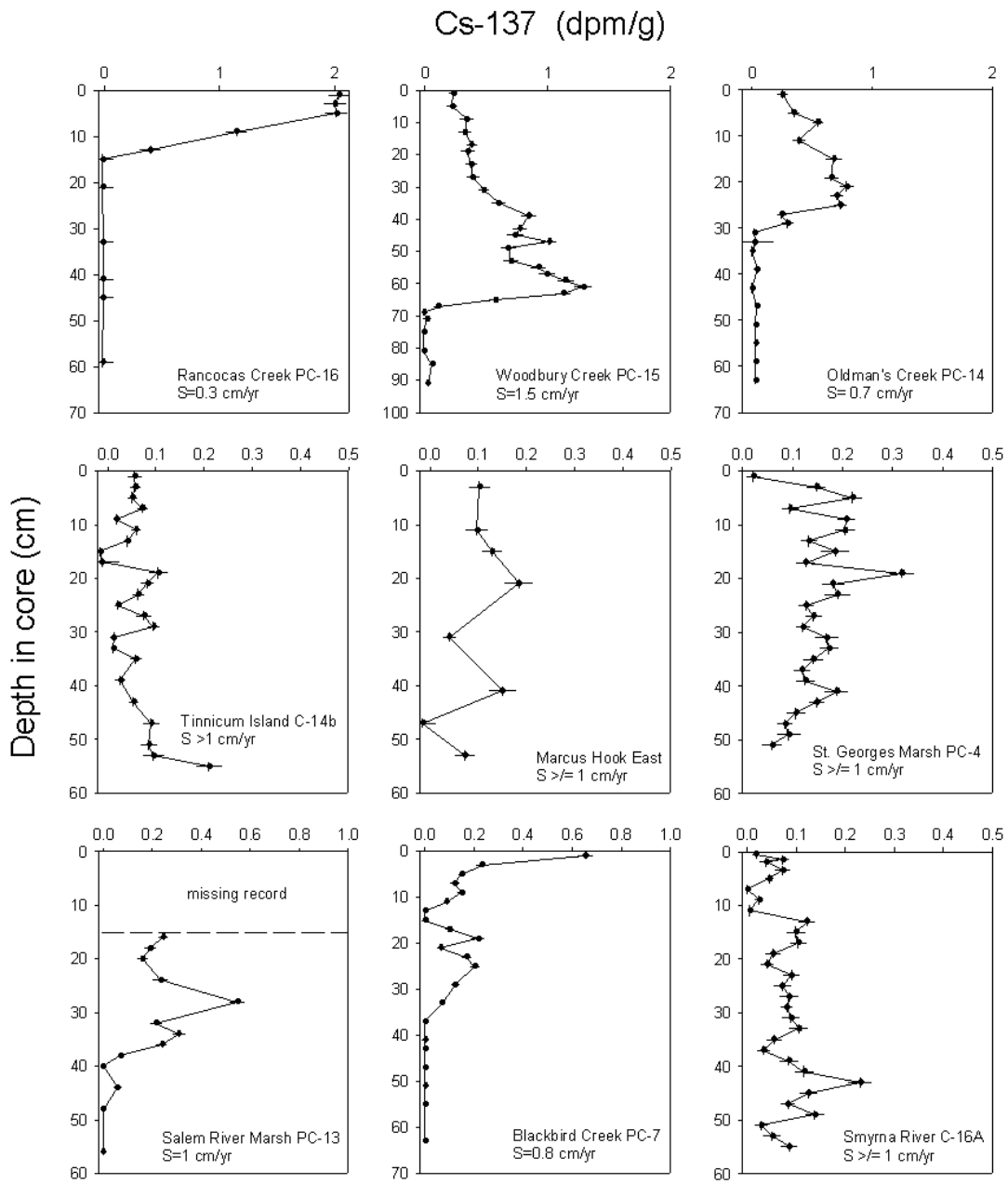

Figure 41. Cs-137 activity profiles for estuary and marsh cores. Note the different depth scales.

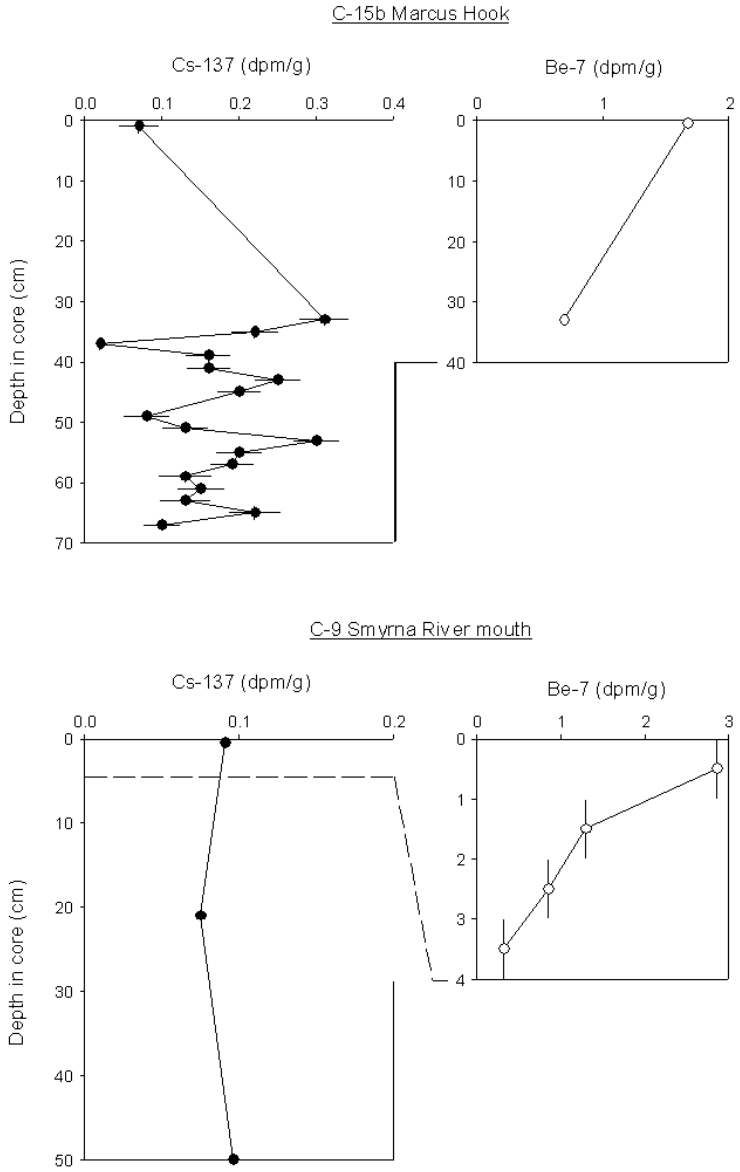

Figure 42. Cs-137 and Be-7 activity profiles from HDC cores. Note difference in depth scales.

Excess Pb-210 profiles for four of the marsh sites exhibited steady-state behavior, shown by a monotonic decrease in activity with depth (Figure 43). Sedimentation rates computed by least-squares linear regression of the activity profile ranged from 0.7 to 1.4 cm/yr. Lead-210 and Cs-137 based rates agreed well for sites PC-13 and PC-14 but differed by nearly a factor of two for PC-15 (Table 5). In part this difference may be attributed to the dissimilar timespans over which these radiotracers average sedimentation rate, in the case of PC-15, 47 years for Cs-137 and 90 years for Pb-210 . Because the Pb-210 geochronology averages rates over a wider range of sedimentary conditions, including periods of non-deposition or minor erosion, it is intuitive that they should be somewhat lower than those based on Cs-137 chronology. An alternative explanation is that the rate of sedimentation at site PC-15 has increased during the past several decades, perhaps due to increasing sediment delivery to the marsh. Although this scenario cannot be ruled out completely, a recent increase in sedimentation rate is not immediately apparent from the radioisotope profiles.

Lead-210 profiles for sites MHE and C14b exhibited variable activities downcore with no net decrease (Figure 43). These non-steady-state profiles cannot be used to compute reliable sedimentation rates, but they support the inference that the bed is physically reworked and that short-term deposition is episodic; similar Pb-210 profiles have been observed in other river-estuaries where bed sediments are known to be continually resuspended and redeposited (Hirschberg et al., 1996). Again, presence of well-preserved bedding structures in these deposits supports the contention that physical processes, rather than bioturbation, are responsible for the shape of the activity profiles.

5.6.4. Radioisotope Inventories

Sediment inventories of radioisotopes (the depth-integrated activity) are useful for establishing how the supply of a radiotracer is partitioned by physical and physiochemical processes within river-estuarine systems. Specifically, the spatial distributions of inventories can elucidate material depocenters that result from sediment "focusing", a process by which suspended sediments (and adsorbed constituents) are preferentially sequestered in some areas over others (Olsen et al., 1993). For example, a

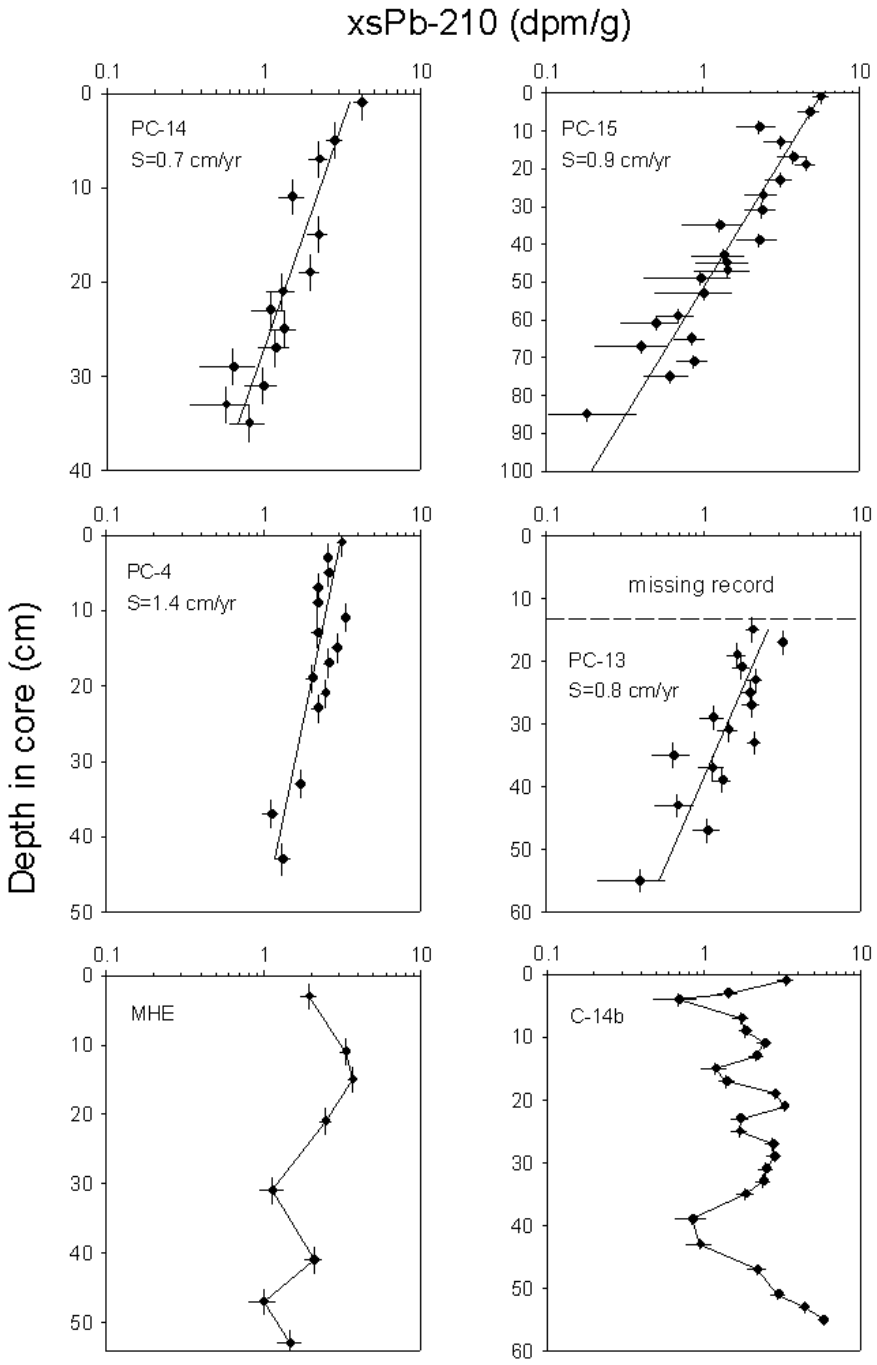

Figure 43. Profiles of excess Pb-210 activity. Note the different depth scales. Steady-state-type profiles are observed for marsh sites PC-4, PC-13, PC-14, and PC-15 (see text), non-steady-state for MHE and C-14b in the open estuary.

Table 5. Sediment accumulation rates and radioisotope inventories

Site	Cs-137 penetration (cm)	Cs-137 maximum (cm)	Cs-137 sed. rate (cm/yr)	Cs-137 inventory (dpm/cm^2)	Pb-210 sed. rate (cm/yr)	xsPb-210 inventory (dpm/cm^2)
C-1	nd[a]	nm[b]	-	-	-	-
C-2	nd	nm	-	-	-	-
C-4	nd	nm	-	-	-	-
C-7	nd	nm	-	-	-	-
C-14b	55	nm	>1.0	3.0	-	-
C-15b	68	nm	-	-	-	-
C-16A	55	43	≥1.0	3.5	-	-
MHE	54	nm	≥1.0	2.0	-	-
PC-4	19	51	>1.0	4.4	1.4	60
PC-7	37	nm	0.8	2.4	-	27
PC-13	40	29	1.0	4.0[c]	0.8	28
PC-14	32	nm	0.7	11.5	0.7	52
PC-15	70	61	1.5	22.0	0.9	78
PC-16	14	nm	0.3	12.0	-	33

[a]nd, non-detectable; [b]nm, no Cs-137 maximum; [c]incomplete inventory (see text)

measured Cs-137 inventory in excess of the expected depositional flux could result from extended scavenging during lateral transport of particles prior to burial, and (or) sources of inventory in addition to the local atmospheric flux. Conversely, inventories less than the expected value could be related to loss through erosion or chemical desorption (Olsen et al., 1982). In the case of tidal marsh deposits the two most important sources of Cs-137 and Pb-210 are (1) atmospheric deposition on exposed marsh surfaces and (2) tidal advection, i.e., inventory derived from open-estuary waters.

Cesium-137 and Pb-210 inventories were computed according to the following:

$$I = \Sigma \rho_s X_i (1 - \phi_i) A_i \qquad (4)$$

where I is the radioisotope inventory (dpm/cm^2), ρ_s the mineral density (2.65 g/cm^3), X is the thickness of the sediment interval i (cm), ϕ is the porosity, and A is the activity at interval i (dpm/g). For cores in which radioisotope measurements were continuous with sediment depth, inventories were summed over the 2-cm intervals. Otherwise, activity and porosity values for omitted intervals were interpolated from adjacent values prior to computing the inventory.

Cesium-137 inventories ranged from 1.8 to 21.8 dpm/cm^2 and generally increased with distance up-estuary from Zone 5 to Zone 2 (Table 5). Inventories for PC-14, PC-15, and PC-16 were respectively 105 %, 52 %, and 57 % of the 1954–1980 atmospheric supply of 21 dpm/cm^2, and are high compared to the Delaware salt marsh (Kim et al., 1997) and US lowlands in general (Graustein and Turekian, 1986). These high inventories likely arise from a combination of high specific activity (due to fine-sediment size or clay mineralogy) and rapid sedimentation rate. Clearly, the freshwater marshes sampled in this study are efficient traps for Cs-137 derived from the local atmospheric flux or through tidal advection. By comparison, sites in the open estuary had Cs-137 inventories merely 10–14 % of the predicted value, where Cs-137 was measurable at all. These low inventories may reflect the coarser grain size of the estuarine sediments (i.e., lower specific activity) and (or) an incomplete post-1954 sedimentary record.

Lead-210 inventories provide further insight into the behavior of particle-reactive substances in the Delaware Estuary. The overall spatial distribution of Pb-210 inventories paralleled that of Cs-137, though the spread between high (78 dpm/cm^2) and low (26 dpm/cm^2) values was somewhat smaller (Table 5). The excess Pb-210 inventory range for the Delaware salt marsh and US lowlands is 25.6–27.7 dpm/cm^2 (Graustein and Turekian, 1986; Kim et al., 1997), 32 dpm/cm^2 being the theoretical steady-state inventory supported by a mean atmospheric flux of 1 dpm/cm^2/yr (Graustein and Turekian, 1986). By comparison, inventories for Oldman's Creek, St. Georges Marsh, and Woodbury Creek are respectively 1.6, 1.9, and 2.4 times the theoretical value. This suggests that Pb-210 inventory in addition to that derived from the atmospheric is sequestered, tidal waters being a probable source. Inventories for Salem Marsh and Blackbird Creek at 27–28 dpm/cm^2 approximate the theoretical value, implying that Pb-210 is supplied by atmospheric deposition alone.

It should be noted that Cs-137 is not an ideal proxy for particle-associated chemical species in general. Although this radioisotope is strongly adsorbed to clay minerals in freshwater environments, in marine waters a larger fraction of dissolved-phase Cs-137 stays in solution as it must compete with cations for sorption sites on suspended particles (Olsen et al., 1982 and references therein). This is relevant to the study area as the sampling locations fall within both oligohaline and mesohaline waters.

On the other hand, Pb-210 is considerably more particle reactive than Cs-137, and the observation that Pb-210 and Cs-137 inventories exhibit similar spatial patterns suggests that the Cs-137 distributions are not solely a product of the physicochemical environment.

Another relevant influence on radioisotope distributions in river-estuarine systems is suspended-sediment concentration. The loading of metals (Fe, Mn, Co, and radioisotopes by relation) tends to be lower in the presence of high suspended-sediment concentrations (Benoit and Rozan, 1999). This is observed in the Delaware estuarine turbidity maximum, where water-column metals concentrations exhibit a regional low (Biggs et al., 1983). Because the tidal marshes down-estuary of Marcus Hook are hydraulically contiguous with waters of the turbidity maximum zone, low radioisotope inventories may reflect low specific activities due to high suspended-sediment concentrations.

In summary, it is clear from the radioisotope geochronologies and inventories that Woodbury Creek, Oldman's Creek, Rancocas Creek, St. George's marsh, and Salem marsh are important material sinks in the upper Delaware Estuary. Because they are situated within a particularly sediment-rich reach, there is great potential for these and adjacent tidal marshes to trap material supplied by tidal waters. Indeed, it is probable that the greater tidal-marsh system constitutes a significant terminal sink for fine-grained sediment derived from the Delaware watershed, as well as down-estuary erosional sources. Given their proximity to industrial centers it is therefore likely that these marshes sequester particle-associated pollutants transported in the tidal Delaware River (e.g., Orson et al., 1992). Detailed studies of sediment transport and deposition within the tidal marshes are needed to elucidate their role as fine-sediment sources and (or) sinks in the greater Delaware River-Estuary system.

6. CONCLUSIONS

The major conclusions of this study are summarized below.

(1) Bottom sediment types in the tidal Delaware River and upper estuary range from mud to gravel and are extremely variable both along- and across-channel. Gravel, sand, and mud weight percentages vary by orders of magnitude, though the across-channel variability of sand and mud increases and decreases, respectively, from DRBC Zone 3 to